

Influence of spatially dependent, modeled soil carbon emission factors on life-cycle greenhouse gas emissions of corn and cellulosic ethanol

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Abstract

Converting land to biofuel feedstock production incurs changes in soil organic carbon (SOC) that can influence biofuel life-cycle greenhouse gas (GHG) emissions. Estimates of these land use change (LUC) and life-cycle GHG emissions affect biofuels' attractiveness and eligibility under a number of renewable fuel policies in the USA and abroad. Modeling was used to refine the spatial resolution and depth extent of domestic estimates of SOC change for land (cropland, cropland pasture, grassland, and forest) conversion scenarios to biofuel crops (corn, corn stover, switchgrass, *Miscanthus*, poplar, and willow) at the county level in the USA. Results show that in most regions, conversions from cropland and cropland pasture to biofuel crops led to neutral or small levels of SOC sequestration, while conversion of grassland and forest generally caused net SOC loss. SOC change results were incorporated into the Greenhouse Gases, Regulated Emissions, and Energy use in Transportation (GREET) model to assess their influence on life-cycle GHG emissions of corn and cellulosic ethanol. Total LUC GHG emissions (g CO₂eq MJ⁻¹) were 2.1–9.3 for corn-, –0.7 for corn stover-, –3.4 to 12.9 for switchgrass-, and –20.1 to –6.2 for *Miscanthus* ethanol; these varied with SOC modeling assumptions applied. Extending the soil depth from 30 to 100 cm affected spatially explicit SOC change and overall LUC GHG emissions; however, the influence on LUC GHG emission estimates was less significant in corn and corn stover than cellulosic feedstocks. Total life-cycle GHG emissions (g CO₂eq MJ⁻¹, 100 cm) were estimated to be 59–66 for corn ethanol, 14 for stover ethanol, 18–26 for switchgrass ethanol, and –7 to –0.6 for *Miscanthus* ethanol. The LUC GHG emissions associated with poplar- and willow-derived ethanol may be higher than that for switchgrass ethanol due to lower biomass yield.

Abbreviations

AEZ = Agro-Ecological Zones
 CCLUB = Carbon Calculator for Land Use Change
 COLE = Carbon On-line Estimator (of U.S. Forest Service)
 CT/RT/NT = Conventional Tillage/Reduce Tillage/No Tillage
 EF = Emission Factor
 GHG = Greenhouse Gas
 GREET = Greenhouse gases, Regulated Emissions, and Energy use in Transportation model (of Argonne National Laboratory)
 GTAP = Global Trade Analysis Project (of Purdue University)
 IPCC = Intergovernmental Panel on Climate Change
 LCA = Life Cycle Analysis
 LCFS = Low Carbon Fuel Standard (of California)
 LUC = Land Use Change
 RED = Renewable Energy Directive (of EU)
 RFS = Renewable Fuel Standard (of USA)
 SOC = Soil Organic Carbon

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BG = billion gallon(s)

BL = billion liter(s)

Gal = gallon(s)

Ha = hectare(s)

L = liter(s)

Mha = million hectare(s)

t = metric ton(s).

Keywords: GREET model, land use change, life cycle analysis, *Miscanthus*, poplar, surrogate CENTURY model, switchgrass, willow

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Introduction

To reduce the greenhouse gas (GHG) intensity of the transportation sector, governments in the USA, the European Union (EU), and elsewhere have developed policies that encourage the production and use of low net GHG-intensity biofuels. As these policies have been developed and implemented, concerns have been raised about their influence on the land resource, both its availability and on its carbon stocks (Fargione *et al.*, 2008; Searchinger *et al.*, 2008; Melillo *et al.*, 2009). Considering the quantities of feedstock required to meet policy targets, these concerns have merit. The current US renewable fuel standard (RFS2) establishes a production volume target for corn (*Zea mays* L.) ethanol of 56.8 billion liters (BL) (15 billion gallons (BG)) by 2015. This fuel volume corresponds to roughly 130 million metric ton (t) of corn (Mumm *et al.*, 2014). Between 2008 and 2014, during which time corn ethanol production increased by 19 BL (RFA (Renewable Fuels Association), 2015), land planted in corn increased by about 2 Mha (Mha) on net. The maximum gain in land planted in corn during this period was 4.5 Mha in 2012 (USDA (U.S. Department of Agriculture), 2015). Although no definitive data exist on the amounts and types of domestic (or international) lands influence by this expansion, grasslands and forests have both likely been affected (Wright & Wimberly, 2013; Lark *et al.*, 2015). Moreover, a significant amount of biomass must be produced to meet the RFS call for the production of 60.6 BL (16 BG) of cellulosic biofuels by 2022 (US Congress, 2007). For example, 188 million t of feedstock would be required to produce the RFS-mandated volume of cellulosic fuels if we assume an ethanol yield from cellulosic feedstocks (e.g., corn stover or energy grasses) of 355 L t⁻¹. A significant amount of land would be needed if this biomass demand were met entirely with energy crops. For example, about 21 Mha of land would be required to produce switchgrass (*Panicum virgatum* L.) and 13 Mha for *Miscanthus* (e.g., *Miscanthus × giganteus*)

if we assume biomass yields of 9 t ha⁻¹ for switchgrass and 15 t ha⁻¹ for *Miscanthus* (US DOE (U.S. Department of Energy), 2011).

Transitioning lands in other uses to biofuel feedstock production will affect the carbon stocks of these lands (Melillo *et al.*, 2009; Fargione *et al.*, 2010). Even if agricultural residues, such as corn stover, are used toward this demand, which would largely change land management rather than land use/cover, soil organic carbon (SOC) stocks will still be affected (Liska *et al.*, 2014; Wu *et al.*, 2015). To achieve decreases in transportation sector GHG emissions, biofuel policies require that the life-cycle GHG emissions of eligible biofuels be less than life-cycle GHG emissions for gasoline. For example, the Renewable Fuel Standard (RFS) mandates that fuels that qualify as cellulosic biofuels achieve a 60% reduction in life-cycle GHG emissions as compared to conventional fossil fuels (EPA (U.S. Environmental Protection Agency), 2010). GHG emissions associated with land-use change (LUC) (i.e., carbon stock changes) stemming from biofuel feedstock production can significantly influence biofuel life-cycle GHG emissions but are difficult to quantify. LUC GHG emissions rely on economic model predictions of the amount, types, and locations of lands that are affected by expansions in biofuel feedstock production, and on associated changes in carbon stocks (e.g., aboveground biomass, soil carbon). Since the inception of the RFS and the California Air Resources Board's (CARB) Low Carbon Fuel Standard (LCFS), great strides have been made both in improving the economic models that estimate LUC and in estimates of carbon stock changes upon LUC. While the former advancement is outside the scope of this article, we describe the progression of techniques used to improve soil carbon stock change estimates, which include improved spatial resolution and modeling techniques, in the supporting information.

Soil carbon modeling is a critical tool that supplements empirical data in the understanding of the effect of feedstock production on soils and in the estimation

of LUC GHG emissions. Earlier SOC estimates in LCA studies have mostly investigated changes in carbon stocks in top soils (20 or 30 cm), and if SOC modeling results are used, they were developed at the regional or national scale. Recent studies, however, suggest that LUC may affect deep (>30 cm) SOC stocks (Knops & Bradley, 2009; Follett *et al.*, 2012; Qin *et al.*, 2015). Also, low-resolution modeling fails to capture SOC spatial heterogeneity that stems from spatially specific climate, soil, and environmental conditions. Efforts to improve SOC modeling for use in LUC GHG estimates must increase the model's ability to quantify the suitability of different land resources for feedstock production from the point of view of improving or maintaining SOC and to determine how increasing the spatial resolution of soil carbon modeling influences the estimation of LUC GHG emissions, which in turn influences biofuel life-cycle GHG emissions.

The objectives of this study are to quantify spatially resolved SOC changes to a 100 cm depth caused by LUC resulting from biofuel feedstock production, and then estimate LUC-associated and life-cycle GHG emissions of corn and cellulosic ethanol. First, we use results from model-derived SOC change estimates to examine how different land transitions will influence SOC changes at a US county level to identify regions where biofuel feedstock production could enhance or degrade SOC stocks. Next, we evaluate the sensitivity of SOC change to key modeling assumptions, which include initial land use, soil depth, energy crop yield, and management practices. Finally, we determine how these variations influence LUC and life-cycle GHG emissions for ethanol produced from corn, corn stover, switchgrass, and *Miscanthus*.

Materials and methods

Modeling SOC changes

Model development. The surrogate CENTURY model (Kwon & Hudson, 2010; Kwon *et al.*, 2013) was used in the study to simulate SOC dynamics in different LUC scenarios (see Supporting Information). It was developed as a nonlinear regression tool that is designed for the parameter estimation of CENTURY (version 4.0)'s SOC dynamics submodel (Kwon & Hudson, 2010). In the model, simulation processes related to CENTURY's mass balance and decomposition kinetics for the three primary soil organic matter pools (i.e., active, slow, and passive) and plant litter pools employ various statistical functions embedded within a nonlinear regression procedure of SAS (SAS Institute, 2011) that provides parameter estimation, simulation, and forecasting of dynamic nonlinear simultaneous equation models. Importantly, the surrogate CENTURY model treats time-dependent model variables (e.g., crop/biomass yield and soil/climate conditions) as model inputs so that the

simulation uncertainties can be isolated and quantified individually (Kwon & Hudson, 2010; Kwon *et al.*, 2013). The C input rates to soils are derived empirically from observed yield/biomass, harvest index (*HI*, the ratio of grain mass to total aboveground plant production of grain crops), root to shoot ratio (*RS*, the ratio of total root production to total aboveground shoot production), and soil temperature/moisture effects on SOC decomposition (Table S1).

In this study, modeling efforts were refined from the state level (Kwon *et al.*, 2013) to the county level to better simulate the temporal and spatial variations of SOC changes associated with four major biofuel feedstocks (i.e., corn, corn stover, switchgrass, and *Miscanthus*) and two short rotation woody biofuel feedstocks, poplar (*Populus* L.) and willow (*Salix* L.). Additionally, the modeled depth was extended from 30 cm to 100 cm to simulate SOC dynamics within two soil layers, topsoil (0–30 cm) and subsoil (30–100 cm), under the influences of county level spatially specific land use types, crop yield, and soil and climate conditions (Fig. S1). The soil carbon pools and flows during decomposition were the same for the topsoil as in Kwon & Hudson (2010), but with additional active, slow, and passive pools for the subsoil. The allocation of belowground biomass and temporal influences of soil and climate conditions were updated for both soil layers (Table S1). More information on the input data, assumptions, model updates and a summary of new coefficients describing poplar and willow allometry, and carbon allocation to the subsoil for all crops is available in the Supporting Information.

LUC and management scenarios. We considered four historical land use types (i.e., cropland, cropland pasture, grassland, and forest) and five biofuel crop production scenarios (i.e., corn, switchgrass, *Miscanthus*, poplar, and willow). Each biofuel feedstock was grown only in counties determined to be suitable (Table S2) based on historical land availability (Kwon *et al.*, 2013) and ability to produce the biofuel crop (Halbleib *et al.*, 2012; Eaton *et al.*, 2013). Land use history was constructed for model spin-up by dividing the entire simulation into: (1) pristine (prior to 1881), (2) pre-modern (1881–1950), and (3) modern (1951–2010) agricultural time periods. Note that cropland pasture is defined as cropland that was in pasture for part of the modern period. We designated the 30-year time period from 2011 to 2040 as the period for feedstock production (Table S2). This is the period over which LUC GHG emissions are amortized, and we refer to it herein as the time horizon. We chose a 30-year time horizon to match that of US EPA and because most of SOC change occurs during the first 30 years following LUC (Qin *et al.*, 2015). For corn, a combination of tillage and residue removal practices was applied in the simulation; this includes three tillage practices, conventional tillage (CT), reduced tillage (RT), and no tillage (NT), and two residue removal levels, 0% and 30%. For the remaining four perennial crops, we assumed these would not be tilled (NT) and modeled them with one residue/biomass removal rate (90%) (Table S2).

Crop yield scenarios. For historical (prior to 2011) and reference yields (2011), we used surveyed crop yield for grain crops (i.e., corn, other grain crops) (USDA, 2015) and modeled

biomass yield for the four perennial biofuel crop scenarios. To predict yields occurring within the 30-year time horizon (2011–2040), we considered (1) constant yield (no annual yield change) and (2) increased yield (annual increase) scenarios. For corn, the constant yield scenario used 20-year county level yield averages from 1991 to 2010; the yield increase was projected using historical yield data (Kwon *et al.*, 2013). For other crops, the county-level reference yields were used for the constant yield scenarios and a 1% annual increase was projected for yield increase scenarios.

A total of 80 model runs, 40 LUC scenarios (Table S2) with two yield scenarios each, were conducted to simulate dynamic SOC (both 30 and 100 cm) for each county (over 2000 counties per LUC scenario). The SOC sequestration rate ($\text{t C ha}^{-1} \text{ year}^{-1}$) was calculated as the annual SOC change during the bioenergy crop production period following LUC (2011–2040), with a positive value indicating net SOC gain and a negative value indicating SOC loss. More information on SOC modeling is provided in Supporting Information.

Estimating LUC and life-cycle GHG emissions

LUC estimations. Figure 1 illustrates the combination of models and data sets that we used to model LUC and

life-cycle GHG emissions. The overall model that we use is called the Greenhouse gases, Regulated Emissions and Energy use in Transportation (GREETTM) model (Argonne National Laboratory, 2014). It has an LUC module called the Carbon Calculator for Land Use Change (CCLUB) (Dunn *et al.*, 2014). CCLUB contains results from modeling runs with an economic model, the Global Trade Analysis Project (GTAP) model (Purdue University, 2015), that predict worldwide LUC upon specific shocks (Table 1) in ethanol production from either corn, corn stover, switchgrass, or *Miscanthus* feedstocks (Taheripour *et al.*, 2011; Taheripour & Tyner, 2013). While shocking the model with production of a single biofuel produced from a single feedstock ignores important interactions between production of different biofuels like biodiesel and production of ethanol from different feedstocks (Kim *et al.*, 2014), it is a technique that permits assignment of all LUC GHG emissions to one feedstock-to-fuel pathway. This approach is used by both EPA and CARB in the RFS and LCFS, respectively. GTAP predicts land shifts between cropland, cropland pasture, grassland, and forest at the Agro-Ecological Zone (AEZ) level. It is important to note that it is not possible to distinguish indirect and direct LUC within the USA from GTAP results. All international LUC could be considered indirect.

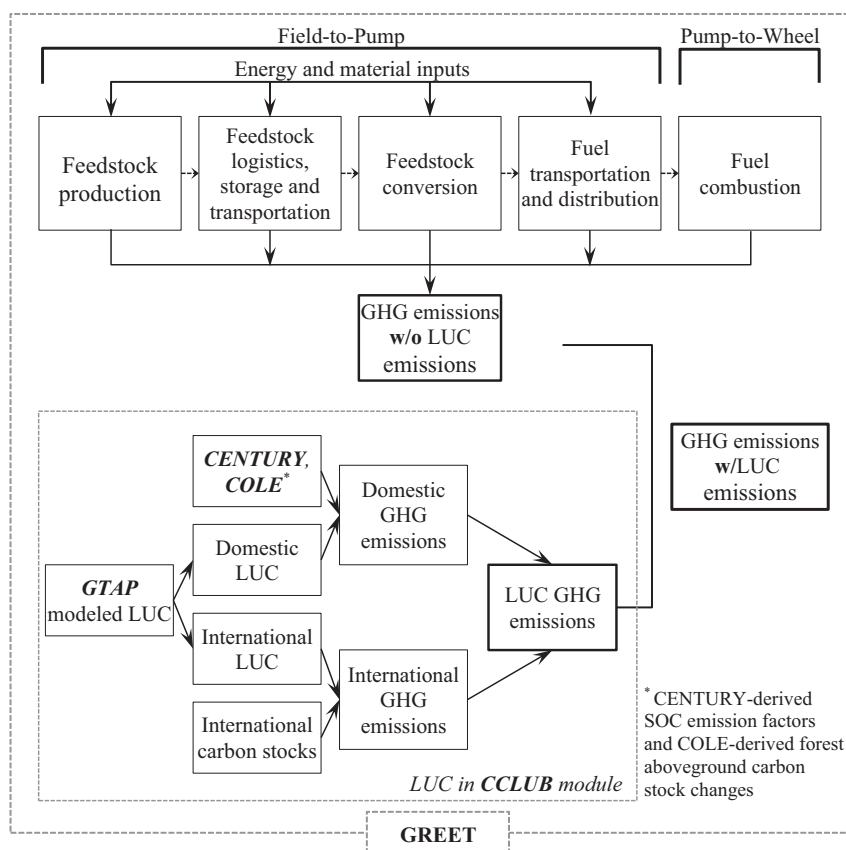


Fig. 1 System boundary in GREET and nested calculation framework for life-cycle GHG modeling with and without land use change (LUC) emissions. GREET models five biofuel life-cycle stages. The CCLUB module contains results and data from several models and sources (Harris *et al.*, 2009; Taheripour *et al.*, 2011; Kwon *et al.*, 2013; Taheripour & Tyner, 2013) (*in bold and italic*) that are used to estimate LUC GHG emissions.

Table 1 GTAP biofuel production scenarios used to simulate LUC as described in CCLUB*

Case	Description	Ethanol increase	Notes and references
Corn ₁	Increase in corn ethanol production from its 2004 level of 13 BL (3.41 BG) to 57 BL (15 BG), using the 2004 database	44 BL	Taheripour <i>et al.</i> (2011)
Corn ₂	Increase in corn ethanol production from its 2004 level (13 BL) to 57 BL, using the 2004 database, with GTAP calibrated CET function	44 BL	Case D of Taheripour & Tyner (2013)
Stover	Increase of ethanol from corn stover 34 BL, on top of 57 BL corn ethanol	34 BL	Taheripour <i>et al.</i> (2011)
Switchgrass	Increase of ethanol from switchgrass by 26.5 BL, on top of 57 BL corn ethanol	26.5 BL	Taheripour <i>et al.</i> (2011)
<i>Miscanthus</i>	Increase of ethanol from <i>Miscanthus</i> by 26.5 BL, on top of 57 BL corn ethanol	26.5 BL	Taheripour <i>et al.</i> (2011)

*Detailed information of CCLUB is available at <https://greet.es.anl.gov> and Dunn *et al.*, 2014.

We considered two scenarios for LUC associated with corn ethanol in Table 1, corn₁ (Taheripour *et al.*, 2011) and corn₂ (Taheripour & Tyner, 2013). The corn₂ scenario includes modifications to GTAP that reflect differences in the ease of land transformation for different regions of the world and the greater cost of converting forest to cropland as compared to converting pastureland to cropland. It is important to note that restrictions on LUC in the RFS, such as a check on expanding agricultural land beyond the 2007 baseline, and the limits on converting forestland to feedstock production (Code of Federal Regulations, 2014) are not reflected in this GTAP modeling. The results, therefore, could be viewed as a worst case example. The carbon stock data indicate carbon changes upon LUC.

LUC and life-cycle GHG emissions. Within CCLUB, in the case of domestic LUC, the carbon stock change data are EFs generated with surrogate CENTURY at the county level. Because GTAP results are at the AEZ level, we pair them with an AEZ-level average of the county-level emission factors. If converted domestic land is forested, county-level forest carbon stocks and foregone sequestration data from Carbon On-line Estimator (COLE) (Van Deusen & Heath, 2010) are applied,

again rolled up to the AEZ level. We match international LUC with carbon stock data from Harris *et al.* (2009) that US EPA used in their analysis. Overall, CCLUB multiplies the GTAP-predicted land area changes by the carbon stock changes to estimate of LUC GHG emissions (Fig. 1).

The LUC GHG emissions are then combined with GHG emissions (e.g., CO₂, N₂O, and CH₄), calculated in the primary GREET model, from the feedstock production, feedstock transportation, conversion, fuel transportation and distribution, and fuel use portions of the biofuel life cycle to produce an overall life-cycle GHG estimate for a given biofuel (Fig. 1). Detailed information regarding CCLUB and GREET modeling can also be found at Dunn *et al.* (2014) and Wang *et al.* (2012), respectively. It should be noted that fertilizer and energy intensity values in GREET (e.g., g N fertilizer t⁻¹ corn, MJ diesel t⁻¹ corn) (Wang *et al.*, 2012) are yield independent and are not influenced by yield data used in SOC modeling. LUC associated with large-scale ethanol production from poplar and willow has not been subject to analysis with GTAP due to the relatively modest contribution to overall biofuel feedstock production (US DOE, 2011). To determine how SOC might respond to a land transition to produce these short rotation woody crops (SRWC) in different areas of USA and expand GREET modeling capacity, we included poplar and willow in our modeling work. We only report life-cycle GHG emissions for ethanol produced from corn, corn stover, switchgrass, and *Miscanthus*. However, because LUC modeling results for cellulosic crops are largely dependent on crop yield, and because SRWC yields fall below switchgrass yields (Fig. S2), we can infer what the bounds on LUC GHG emissions for SRWC ethanol could be and discuss these qualitatively.

Results

Domestic rates of SOC change

Refinement of modeling scale. Evaluation of county-level emissions suggests that sequestration rates can vary greatly among LUC scenarios and counties within the same region (Fig. 2). For cropland and cropland pasture conversions, the areas with high SOC gain (e.g., Fig. 2) are normally those with high crop yield (e.g., Fig. S2). For example, in cropland conversions, the Midwest for corn (Fig. 2a), *Miscanthus* (Fig. 2i) and willow (Fig. 2q), the southeast for switchgrass (Fig. 2e), and the east for poplar (Fig. 2m) are regions with higher SOC sequestration rates and relatively higher crop yields (Fig. S2). For SOC changes upon forest and grassland transitions, the spatial trends in SOC sequestration rates appeared to be less closely related to yield trends and the influence of yield was more variable. The areas that gain SOC are those where energy grass yields are expected to be high, such as the southeast region for switchgrass and the Midwest for *Miscanthus* (Fig. S2), converting grassland to switchgrass (Fig. 2g) or *Miscanthus* (Fig. 2k) could increase SOC. Overall, where grassland or forest is

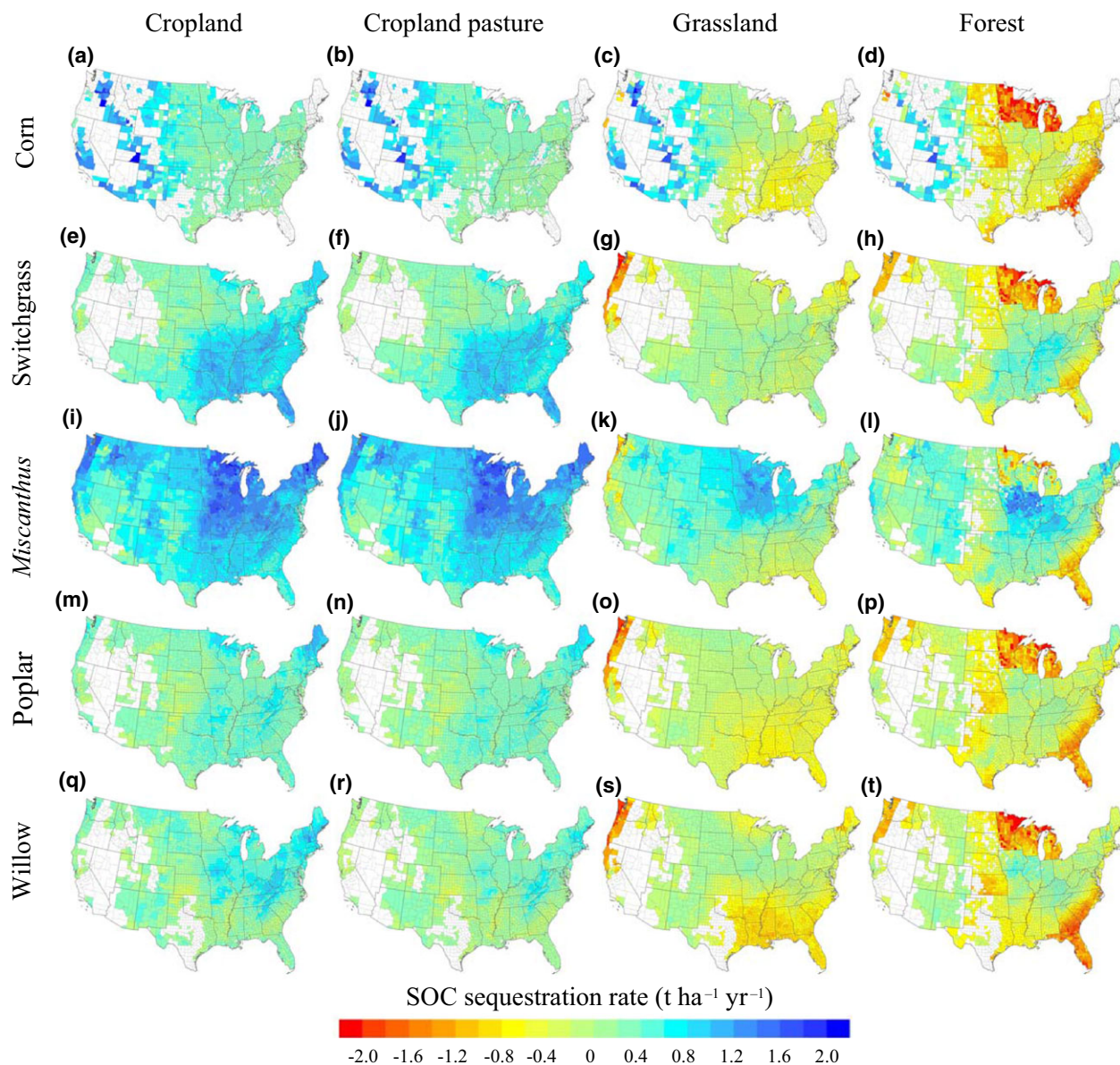


Fig. 2 Modeled soil organic carbon (SOC) sequestration rates at 100 cm soil depth for land use changes from cropland, cropland pasture, grassland and forest to corn, switchgrass, *Miscanthus*, poplar, and willow with yield increase. Corn was averaged over different tillage practices and residue removal rates. The SOC sequestration rates were based on a 30-year time horizon. A positive value indicates net carbon gain while a negative value indicates carbon loss. Note that the rates are not weighted by crop area.

converted to production of bioenergy crops where yields are low, it is likely SOC will decline (e.g., Fig. 2c and d). It is worth noting that high corn yield contributed significantly to the SOC increase in most cropland conversions and even some grassland/forest conversions (Fig. 2). Particularly, in some areas of the western USA (e.g., Arizona, California, Idaho, Oregon, Utah, and Washington), the historical corn yield (2001–2010) is about 10% (Utah) to over 40% (Washington) higher than the rest of the USA (USDA, 2015), mainly due to the widespread use of irrigation (over 90% of corn grown in these states)

(USDA, 2012). These regions, however, are not high corn producers due to their extremely limited corn areas (total harvested area of 2006–2010 accounts for only 2.4% of the national total) (Fig. S7). Future models (e.g., LUC modeling) will need to further restrict pools of land available for LUC based on supporting resources (e.g., water).

When we average emissions over all counties to evaluate the general effect of LUC on GHG emissions, we reconfirm that in general, conversions from cropland and cropland pasture to corn would lead to net SOC

sequestration, and that converting grassland and forest to corn would cause net SOC loss in most of the scenarios (Fig. 3). Median rates upon transition to corn systems showed overall SOC increases of $0.1\text{--}0.4\text{ t C ha}^{-1}\text{ year}^{-1}$ when the initial land use was cropland (Fig. 3a) or cropland pasture (Fig. 3b). Cropland pasture conversions had lower predicted SOC sequestration rates than corresponding cropland conversions because they had higher reference SOC levels inherited from inclusion of deeper rooted perennial crops within the rotation in the recent past (Table S2). When the initial land use was grassland (Fig. 3c) or forest (Fig. 3d), SOC decreased -0.4 to $-0.1\text{ t C ha}^{-1}\text{ year}^{-1}$ and -0.7 to $-0.4\text{ t C ha}^{-1}\text{ year}^{-1}$, respectively. Regardless of the initial land use state, when land transitioned to production of cellulosic crops (i.e., switchgrass and *Miscanthus*), the averaged SOC seques-

tration rates showed net carbon sequestration in most counties in almost all LUC scenarios (Fig. 4). New modeling scenarios suggest converting cropland and cropland pasture to woody crops (i.e., poplar and willow) resulted in net SOC sequestration in most cases (Fig. 4a and b), and conversion of grassland or forest is likely to lead to overall SOC loss for both systems (Fig. 4c and d).

SOC responding to yield and management factors. Because yield is a major determinant of SOC dynamics, we used a relatively conservative assumption to simulate the annual yield with constant (Figs S4 and S5) and increase scenarios (Figs 3 and 4). The SOC sequestration rates under constant yield were generally lower, and the difference between the constant and increasing yield scenarios was most significant (30% greater) for crop-

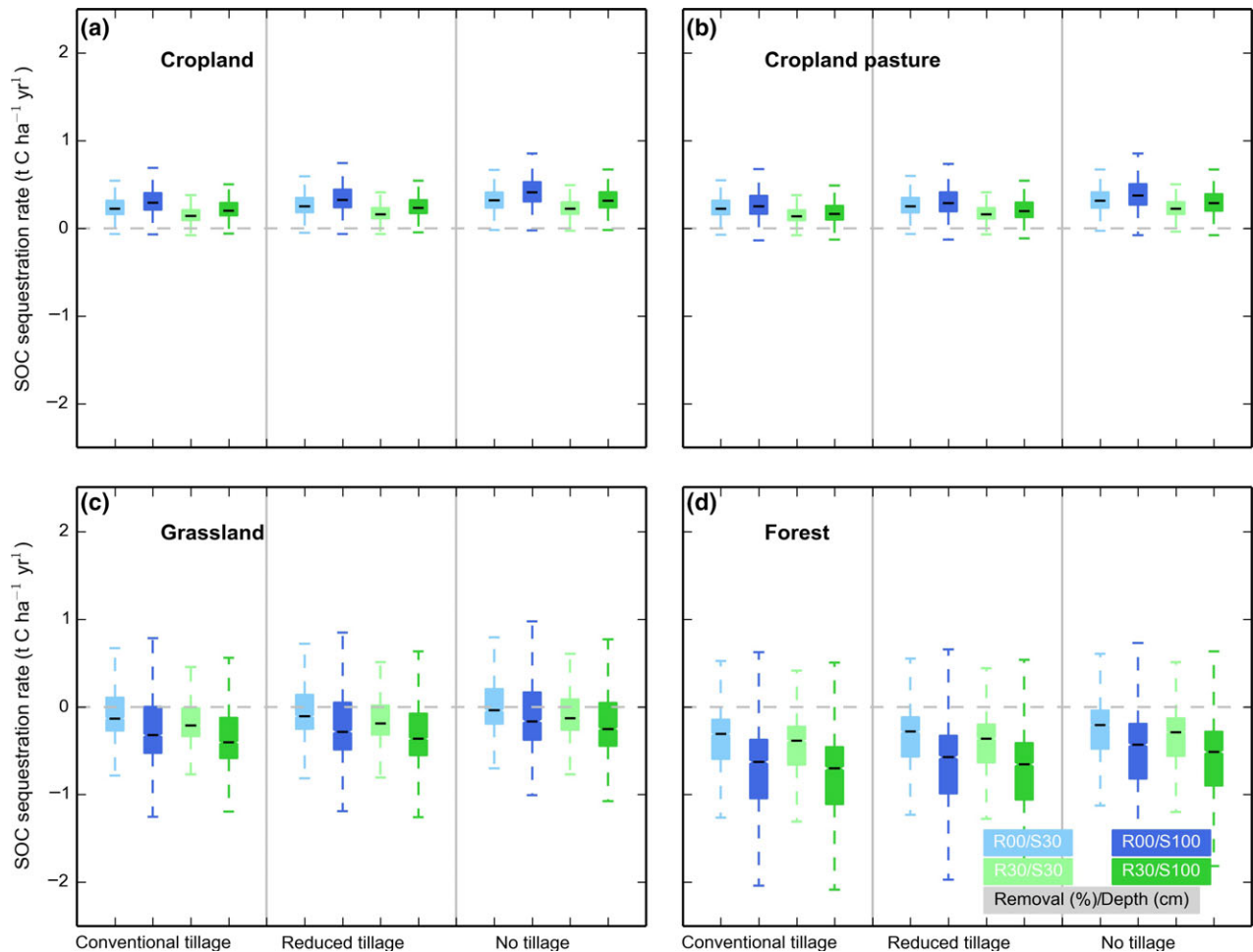


Fig. 3 Soil organic carbon sequestration rates in the 30 and 100 cm soils for land use changes from (a) cropland, (b) cropland pasture, (c) grassland and (d) forest to corn with yield increase under different management practices. R00, R30 indicate 0% and 30% residue removal rates, respectively; S30 and S100 indicate 0–30 and 0–100 cm soil depths, respectively. The box plot for all counties considered in Table S2 (same in Figure 4): the bottom and top of the box are the first and third quartiles, and the band inside the box is the second quartile (the median). The ends of the whiskers represent 1.5 IQR (interquartile range).

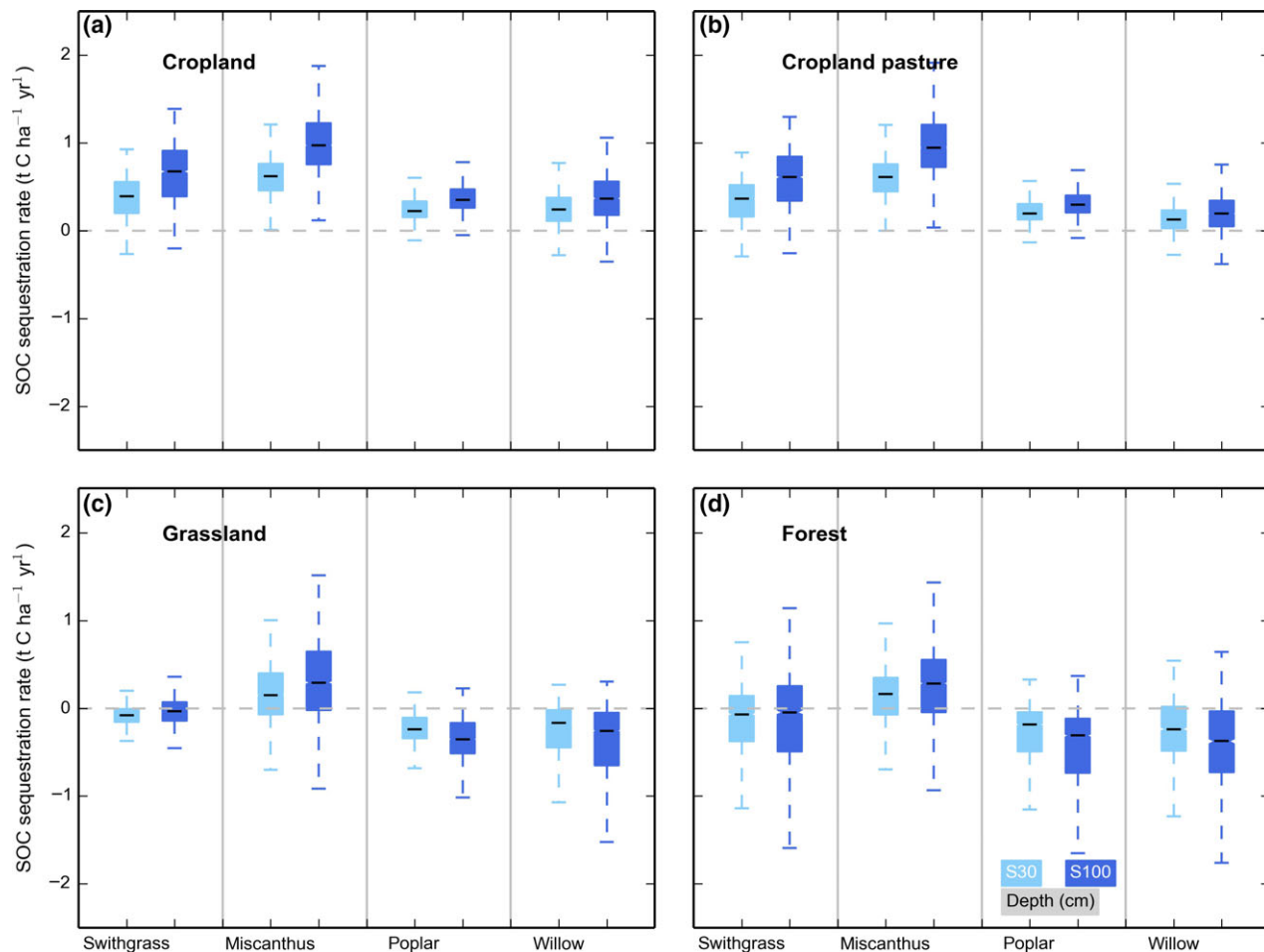


Fig. 4 Soil organic carbon sequestration rates in the 30 and 100 cm soils for land use changes from (a) cropland, (b) cropland pasture, (c) grassland, and (d) forest to switchgrass, *Miscanthus*, poplar, and willow with yield increase. S30 and S100 indicate 0–30 and 0–100 cm soil depths, respectively.

land and cropland pasture conversions. In the case of cellulosic and woody crop systems, this difference was 10%, regardless of soil depth. For grassland converted to corn, the difference was about 15%. For all other cases, the differences varied among LUC scenarios from 0 to 10%.

The magnitude of SOC change upon transitions to corn agriculture also depended on residue removal rates and tillage practices. While other factors (i.e., tillage practices and soil depths) were fixed, residue removal resulted in less SOC sequestration regardless of the initial land use (Fig. 3). For land transitions from cropland and cropland pasture to corn, the SOC sequestration rates under 30% residue removal were about 82–86% of those under no residue removal. Generally, less frequent tillage resulted in higher positive SOC sequestration rates in cropland (Fig. 3a) and cropland pasture (Fig. 3b).

In grassland and forest conversions, residue removal had a less notable effect on SOC change than seen in

cropland transitions; the difference in sequestration rates between residue removal levels of 0% and 30% was 8–12% for grassland transitions to corn and only 2–5% for forest transitions (Fig. 3b). For these conversions, environmental factors (i.e., climate, soil conditions) may exert more influence on SOC change than seen in cropland conversion scenarios.

SOC sequestration in different soil depths. Extending the depth of soil considered increased the magnitude of the SOC change, but did not affect the overall direction of change for different LUC scenarios. For land transitions to corn systems, compared with 0–30 cm soils, the absolute SOC sequestration rates in 0–100 cm soils were 5–30% larger if initial lands were from cropland (Fig. 3a) and cropland pasture (Fig. 3b), and about 30–50% larger if initially converted from grassland (Fig. 3c) and forest (Fig. 3d). The trends also apply to cellulosic and wood crop systems (Fig. 4). In particular, when cropland is

converted to switchgrass or *Miscanthus* (Fig. 4a), the deeper soils can sequester up to 60% more carbon than the surface 30 cm soils. The amplified SOC change in deeper soils was most apparent for cropland transitions (Fig. 4a) mostly because conventional crops (e.g., corn and wheat) have shallower roots than cellulosic and woody crops.

Estimating SOC change and GHG emissions

The AEZ-level SOC sequestration rates showed a similar pattern of SOC changes as indicated above (Fig. S6). Conversions of cropland and cropland pasture generally reinforce soil carbon sinks, while replacing grassland and forest with biofuel crops is very likely to reduce the SOC content in most of the cases. It should be noted that high SOC sequestration rates do not necessarily mean high levels of SOC change within identified regions. To determine potential for sequestration associated with LUC, the affected area within specific land use types (Fig. S7) is used in the calculation of total carbon stocks change. In LUC GHG emission estimations, some AEZs (e.g., AEZ14–16) have fewer counties experiencing LUC than others in the main crop growing zones (e.g., AEZ10 and 11) (Fig. S7), and therefore, their contribution to the overall LUC GHG emission estimation is minimal in the context of national accounting. The AEZ-level EFs were used to calculate domestic LUC GHG emissions in CCLUB (Fig. 1). The international LUC GHG emissions calculated with the EFs from Harris *et al.* (2009) are unaffected by varying assumptions about management practices and other aspects of domestic feedstock production that influence surrogate CENTURY model results (Fig. 5).

Land use change GHG emission estimates showed large variations considering all possible scenarios (e.g., constant and increasing yield, all tillage practices, and all residue removal rates) with the total (domestic and international) LUC GHG emissions ranging from $-20.1 \text{ g CO}_2\text{eq MJ}^{-1}$ for *Miscanthus* to $9.3 \text{ g CO}_2\text{eq MJ}^{-1}$ for corn (Fig. 5). Soil depth did not significantly change the LUC GHG emissions for corn and corn stover ethanol, but affected the estimates for cellulosic ethanol (Table 2). For switchgrass, the domestic LUC impacts changed from net carbon sources (0–30 cm) to sinks (0–100 cm) because SOC sequestration was sufficient to offset GHG emissions from farming activities, feedstock transportation, and feedstock conversion when the deeper depth was considered. *Miscanthus* ethanol became a net carbon sink (Table 2) when the deeper soil depth was considered because soil carbon sequestration doubled.

Land use change associated with *Miscanthus* and switchgrass ethanol offered remarkably high GHG mit-

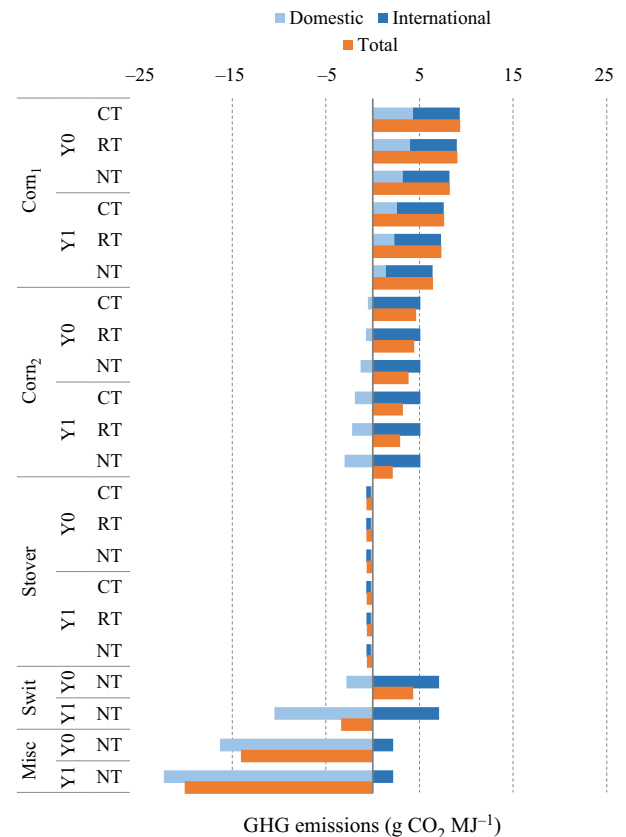


Fig. 5 Summary of CCLUB estimates of domestic, international, and total greenhouse gas emissions derived from land-use change driven by biofuel shocks satisfied by individual biofuel sources, where contributions from domestic soil organic carbon change were assessed to 100 cm. Y0 and Y1 indicate yield constant and increase, respectively. CT, RT, and NT indicate conventional, reduced, and no tillage, respectively.

igation potentials in scenarios that assume annual increase in yield, providing significant carbon sequestration on domestic lands. Yield increase is a major driver of SOC sequestration estimates in these LUC scenarios for these two energy grasses. For corn stover, the GHG emissions from domestic LUC, international LUC, and total were all estimated to be marginally negative (Fig. 5) because GTAP predicts a small amount of gains in forest lands that result in carbon sequestration, offsetting carbon emissions from limited conversion of cropland pasture to corn agriculture. With this small amount of affected land, yield increase and tillage practice assumptions for corn agriculture with stover removal minimally influenced LUC GHG emissions results for corn stover ethanol. Overall, LUC GHG emissions for corn stover ethanol could be considered negligible.

For corn ethanol, we generated GHG emissions for LUC predicted from two different GTAP runs, corn₁

Table 2 Estimated LUC and life-cycle GHG emissions for ethanol produced from corn and cellulosic feedstocks

	Land use change GHG emissions (g CO ₂ eq MJ ⁻¹)			Life-cycle GHG w/o LUC (g CO ₂ eq MJ ⁻¹)‡	Life-cycle GHG w/LUC (g CO ₂ eq MJ ⁻¹)
	Domestic*	International	Total		
Corn ₁	1.4 to 4.3 (1.5 to 3.7)†	5.0	6.4 to 9.3 (6.5 to 8.7)	57.1	63.5 to 66.4 (63.6 to 65.8)
Corn ₂	−3.0 to −0.5 (−2.4 to −0.4)	5.1	2.1 to 4.6 (2.7 to 4.7)	57.1	59.2 to 61.7 (59.8 to 61.8)
Corn stover	−0.2 (−0.2)	−0.5	−0.7 (−0.7)	14.8	14.1 (14.1)
Switchgrass	−10.5 to −2.8 (0.5 to 5.8)	7.1	−3.4 to 4.3 (7.6 to 12.9)	21.8	18.4 to 26.1 (29.4 to 34.7)
<i>Miscanthus</i>	−22.3 to −16.3 (−12.5 to −8.4)	2.2	−20.1 to −14.1 (−10.4 to −6.2)	13.5	−6.7 to −0.6 (3.1 to 7.3)
Gasoline ‡					94

*Range of results with 100 cm and †30 cm (in parentheses) soil depth is produced from SOC modeling scenarios with different yield (constant and increasing) and tillage scenarios (i.e., NT, RT and CT).

‡GREET 2014 (Argonne National Laboratory, 2014) parameters for neat gasoline without blended ethanol.

and corn₂. As described earlier, the corn₂ scenario (Taheripour & Tyner, 2013) incorporated different ease of land transformation rates for different areas of the world and higher costs for conversion of forest to cropland than for conversion of cropland pasture to cropland. Changes in SOC driven by domestic LUC stemming from increased corn production contributed relatively little to total emissions regardless of corn ethanol scenario (Table 2). In the corn₁ scenario, domestic LUC GHG emissions account for 2–6% of total life-cycle GHG emissions, inclusive of LUC GHG emissions. In the corn₂ scenario, domestic LUC GHG emissions could offset up to 5% of life-cycle emissions prior to inclusion of LUC GHG emissions. Total LUC GHG emissions associated with the corn stover ethanol scenario reduce total life-cycle GHG emissions by approximately 2% (Table 2). For cellulosic crops, SOC change is a more important contributor to life-cycle GHG emissions and in fact domestic LUC results in carbon sequestration, reducing life-cycle GHG emissions. For example, domestic LUC GHG emissions in the case of switchgrass ethanol decrease life-cycle GHG emissions (prior to inclusion of LUC GHG emissions) by 13–48%. The large domestic LUC GHG emission reductions in the *Miscanthus* ethanol pathway can offset all GHG emissions elsewhere in the fuel's life cycle and provide additional carbon credits (Table 2).

In the case of corn ethanol, the most important factor determining LUC GHG emissions is the amount of forest predicted to be converted to corn. In LUC estimates from the updated GTAP configuration, Taheripour & Tyner (2013) observed more cropland switching to produce enough corn to meet the mandated level of corn ethanol production. For the corn₁ case (Taheripour *et al.*, 2011), both domestic and international LUC GHG emissions were positive. These emissions were lowest when results from SOC modeling runs assumed corn

yield increase and no tillage. The greatest emissions occurred when we assumed constant corn yield and conventional tillage. It is important to note that provisions of the RFS limit the likelihood of expansion of biofuel crop production on nonagricultural and forested lands and should prevent SOC loss from forest-to-biofuel feedstock production LUC scenarios (Fig. 2d,h,l,p, and t) that could occur or at least prevent an extensive amount of forest conversion to crop production. For example, under the US EPA's aggregate compliance approach (Code of Federal Regulations, 2014), if the amount of agricultural land expands beyond 2007 levels (163 million eligible crop hectares), biofuel producers must show that the land from which feedstock is produced was cleared or cultivated prior to December 19, 2007. Additionally, EPA requires that the land be managed, fallow, and non-forested on December 19, 2007. Recently, however, Lark *et al.* (2015) presented an analysis that indicates declines in grassland and forested lands that may be attributable in part to the expansion of corn agriculture despite the policy instruments in the RFS to limit these LUC types.

Total life-cycle GHG emissions of each type of ethanol (including LUC GHG emissions) were lower than those of conventional gasoline (Table 2). Ethanol from all feedstocks except *Miscanthus* acted as net CO₂ sources, with life-cycle GHG emissions that ranged from 14 g CO₂eq MJ⁻¹ for corn stover ethanol to over 60 g CO₂eq MJ⁻¹ for corn ethanol in corn₁ scenario. The *Miscanthus*-sourced ethanol can be either a net carbon sink (0–100 cm soil) or source (0–30 cm soil) depending on the selected soil depth. In the case of corn ethanol, total LUC GHG emissions are about 5–10% of life-cycle GHG emissions. For corn stover, *Miscanthus* and switchgrass ethanol, in some cases, carbon sequestration from LUC can actually mitigate GHG emissions from processes in the fuels' life cycle (e.g., fuel conversion, transportation) (Table 2).

Discussion

The diverse, spatially explicit results in Fig. 2 highlight the advances possible in estimating changes in soil carbon upon land transitions when spatially explicit carbon stock data are used. Considering the Corn Belt as an example, Fig. 2 clearly shows the wide range of SOC EFs that this region would experience upon transitions to biofuel feedstock production. In contrast, if the IPCC (Intergovernmental Panel on Climate Change) Tier 1 approach (see Supporting Information for a description of this approach) were taken to estimating SOC EFs for this region, the spatial heterogeneity would not be fully captured. The IPCC Tier 1 approach characterizes the Corn Belt with only two dominant climate zones, warm temperate moist and cool temperate moist (IPCC (Intergovernmental Panel on Climate Change), 2006). For each land use management scenario, a limited number of SOC EFs for this region would be calculated from the default reference SOC stocks determined by soil type and the two climate zones. Rather than using a Tier 1-based coarse estimation of SOC change, when possible, process-based SOC modeling should be exercised to accurately capture SOC changes responding to spatially explicit climate, soil, land use, and management factors.

The overall SOC change estimates in this study are in accordance with many previous reports. Many studies have noticed that grassland and forest generally have much higher SOC content, and therefore, the conversions of such lands to managed crop systems inevitably disturb the relatively stable soil system and cause soil carbon loss (Post & Kwon, 2000; Guo & Gifford, 2002; Don *et al.*, 2011; Qin *et al.*, 2015). Using a meta-analysis, Guo & Gifford (2002) observed SOC decreases of about 42–59% and 10–13% when native forest or pasture were converted to field crops and woody plantations, respectively. Qin *et al.* (2015) reported 9–35% carbon losses when grassland and forest were converted to corn and poplar. However, for cropland conversions to certain cellulosic and woody cropping systems, LUC may not necessarily lead to SOC loss. For instance, Guo & Gifford (2002) noticed about an 18% carbon gain in crop-to-woody plantation transitions. In Qin *et al.* (2015), an overall SOC increase of 6–14% was observed in cropland conversions to biofuel crops, while land transitions from forest and grassland to switchgrass, *Miscanthus* and willow did not significantly change SOC. Increased SOC from cropland conversions to corn especially under reduced tillage is consistent with Clay *et al.* (2015) who documented how tillage practice changes and higher yields increased SOC by 24% since 1985 in the Western Corn Belt. Don *et al.* (2011) analyzed LUC in Europe and found that perennial energy crops grown

on former cropland may sequester an additional 0.44 t C ha⁻¹ year⁻¹ SOC for poplar and willow and 0.66 t C ha⁻¹ year⁻¹ for *Miscanthus*. Modeling studies suggested that converting croplands to cellulosic biofuel crops (e.g., switchgrass, *Miscanthus*) can potentially sequester 0.2–0.9 t C ha⁻¹ year⁻¹, depending on crop species and local environment (Qin *et al.*, 2012; Mishra *et al.*, 2013). Kwon *et al.* (2013) used an earlier version of the surrogate CENTURY model and estimated state-level top 30 cm SOC changes under LUCs; they found that converting croplands to corn, switchgrass, and *Miscanthus* increased SOC by about 0.05–0.54 t C ha⁻¹ year⁻¹, while conversions of grassland and forest can cause carbon loss to corn system and neutral to median SOC gain to switchgrass and *Miscanthus*. The sequestration rates predicted in the modeling were generally consistent with corresponding estimates in this study.

The LUC GHG emissions estimated here (Table 2) are comparable with our previous results for corresponding biofuels (Dunn *et al.*, 2013). One source of the variation in results is using SOC EFs at a higher spatial resolution and from deeper soils. For cellulosic ethanol, another source of variation is updates to the process that converts biomass to cellulosic ethanol since the 2013 version of GREET, including updates on ethanol yields and consumption of pretreatment chemicals (Argonne National Laboratory, 2013). The differences between the two cases for corn ethanol in Table 2, caused solely by adjustments to GTAP, highlight the critical role the economic modeling results play in determining LUC GHG emissions. The case using the updated GTAP version has lower LUC GHG emissions than the previous estimates (4.7–11 g CO₂eq MJ⁻¹) (Dunn *et al.*, 2013) regardless of assumptions used in the SOC modeling. Additionally, both corn ethanol scenarios estimated LUC GHG emissions well below Searchinger *et al.* (2008)'s result of 104 g CO₂eq MJ⁻¹. Worst-case corn ethanol life-cycle GHG emissions in Table 2 (66.4 g CO₂eq MJ⁻¹) yield a GHG reduction compared to gasoline of about 30%, which exceeds the RFS GHG reduction threshold for renewable fuels. The cellulosic feedstocks all surpass the RFS 60% GHG reduction target for cellulosic biofuels (Table 2). As described above, we have not conducted LUC modeling for scenarios in which poplar or willow (instead of *Miscanthus* or switchgrass) are used as feedstocks to produce cellulosic ethanol. As LUC model results are a strong function of yield (Dunn *et al.*, 2013) and SRWC yield is fairly close to switchgrass yield in most locations, LUC modeling results for the switchgrass ethanol scenarios can shed light on what we might expect results for poplar- or willow-derived ethanol to be. Given that Fig. 4 shows that conversion of any land type to poplar or willow as compared to switchgrass

either sequesters less SOC (e.g., Fig. 4b) or causes carbon emissions (Fig. 4d), we expect that LUC GHG emissions associated with poplar- and willow-derived ethanol would be higher than that for switchgrass-derived ethanol.

Given the importance of LUC GHG emissions in biofuel policies and because they are an indication of possible effects of biofuel production, future work is still required to improve techniques used to estimate these emissions. Our focus is on improving modeling of SOC changes and examining emerging issues in their application to LUC GHG emissions estimation. One important area of analysis is SOC changes upon corn stover harvest removal from corn fields for use as a biofuel feedstock. This feedstock is already in use at biorefineries in the USA, for example, in Iowa and Kansas. A good deal of work has been done to examine a sustainable stover removal rate that maintains SOC levels (Muth *et al.*, 2013; Karlen & Johnson, 2014). Nonetheless, some have raised concerns that when stover is removed from soils, SOC levels would be lower than if stover were not removed and that this difference between SOC levels with and without stover removal is essentially a carbon debt that biofuels derived from corn stover could incur (e.g., Liska *et al.*, 2014). This 'alternative baseline' approach to calculate GHG emissions from SOC changes differs from estimating GHG emissions or sequestration as the difference between SOC levels at the start and finish of the time horizon. The use of alternative land use as a baseline may only be reasonable if the alternative system is practical, verifiable, and likely (Robertson *et al.*, 2014; Sheehan *et al.*, 2014). Methodology considerations aside, there is a need for a better understanding of spatially explicit SOC changes upon stover removal and the role carbon input techniques such as cover crops and manure addition could have on maintaining and even bolstering SOC levels (Ugarte *et al.*, 2014; Lu, 2015).

Additional future improvements in SOC modeling related to biofuel crop production include the treatment of nitrogen dynamics, irrigation, and modeling time horizon. Nitrogen dynamics can be introduced into the surrogate CENTURY model for the purposes of both estimating spatially explicit N₂O emissions and accounting for feedback between soil carbon and nitrogen (Huang *et al.*, 2009; Melillo *et al.*, 2009). Irrigation, as a possible water input, impacts LUC prediction (Taheripour *et al.*, 2013), biomass production, soil carbon dynamics, and overall GHG emissions from the ecosystem (Le *et al.*, 2011; Qin & Zhuang, 2014) and therefore should be incorporated in the LUC and SOC modeling when spatial data become available. In the study, a 30-year period was used to match the time horizon of U.S. EPA LUC GHG emissions calculations.

However, it should be noted that this may not accurately reflect the agricultural practices for production of corn and other biofuel crops. For example, corn may be rotated with other crops to avoid a possible yield penalty (Gentry *et al.*, 2013), and energy grasses and SRWC may be rotated or replanted during the 30-year time period. A temporally and spatially specific LUC modeling matched with SOC modeling could improve the representativeness of potential cropping activities. Finally, additional efforts are still needed to estimate potential land use change associated with SRWC production, so that LUC GHG emissions can be accurately assessed. Estimating uncertainty associated with LUC GHG emissions is challenging because these emissions are estimated from a combination of results derived from (1) economic modeling, in which error is associated with yield response to price movements and many other parameters and (2) soil carbon modeling, in which error is associated with input data, model parameters and spatial aggregation. At times, especially for emerging feedstocks like poplar or willow, few data exist, making estimation of uncertainty associated with SOC model inputs such as allocation and return extremely difficult if not impossible (Table S1). Uncertainty estimation will improve as more data become available for these crops. Furthermore, our ongoing research aims to explore major uncertainties associated with factors in LUC modeling and the surrogate CENTURY model (e.g., crop yield). We plan to assess the relative magnitude of uncertainty from different elements in LUC GHG emission estimates (e.g., LUC projection, SOC modeling, and GHG emission estimation). Uncertainties associated with GHG emissions from other steps in the life cycle of ethanol besides LUC (e.g., fertilizer consumption) have been identified and quantified in an earlier study (Wang *et al.*, 2012).

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Supporting Information

Additional Supporting Information may be found in the online version of this article:

Model S1. Development of methods to quantify biofuel-induced land use change (LUC) and associated GHG emissions.

Model S2. Incorporating LUC GHG emissions into biofuel life-cycle GHG estimates.

Model S3. The surrogate CENTURY and SOC modeling.

Table S1. Key parameters used in the SOC modeling to simulate impacts of biomass return, tillage and climate on soil carbon dynamics.

Table S2. Land use change scenarios simulated in the SOC modeling.

Figure S1. Agro-Ecological Zones (AEZ) in the United States classified at county-level.

Figure S2. Initial yield (2011) distribution for biofuel crops used in the SOC modeling: (a) corn, (b) switchgrass, (c) *Miscanthus*, (d) poplar and (e) willow.

Figure S3. Modeled SOC sequestration rates at 30 cm soil depth for LUCs from cropland, cropland pasture, grassland and forest to corn, switchgrass, *Miscanthus*, poplar and willow with yield increase. Corn was averaged over different tillage practices and residue removal rates. The SOC sequestration rates were based on 30-yr conversion time period. A positive value indicates net carbon gain while a negative value indicates carbon loss.

Figure S4. SOC sequestration rates in the 30 and 100 cm soils for LUCs from (a) cropland, (b) cropland pasture, (c) grassland and (d) forest to corn with constant yield under different management practices. R00, R30 indicate 0% and 30% residue removal rates; S30 and S100 indicate 0–30 and 0–100 cm soil depths. The box plot (same in Figure S5): the bottom and top of the box are the first and third quartiles, and the band inside the box is the second quartile (the median). The ends of the whiskers represent 1.5 IQR (interquartile range).

Figure S5. SOC sequestration rates in the 30 and 100 cm soils for LUCs from (a) cropland, (b) cropland pasture, (c) grassland and (d) forest to switchgrass, *Miscanthus*, poplar and willow with constant yield. S30 and S100 indicate 0–30 and 0–100 cm soil depths, respectively.

Figure S6. AEZ level SOC sequestration rates at top 30 and 100 cm soils for LUCs from cropland, cropland pasture, grassland and forest to corn (corn), corn stover (stov), switchgrass (swit), and *Miscanthus* (misc) with yield increase. The rates were averaged (arithmetic means) over counties in the AEZ region, with error bars showing the standard deviations. Labels (a) to (h) indicate AEZ 7 to 14, respectively. AEZ 15 and 16 were not presented here, having either very limited or no counties with certain LUCs.

Figure S7. The acreage of harvested corn grain (2006–2010) by state and by AEZ. The primary axis shows the area (Million acres) in each state by AEZ, and the secondary axis indicates the cumulative percentage of the total area of each state. The bar chart indicates the overall distribution of corn area in each AEZ. Data were extracted from USDA (U.S. Department of Agriculture) (2015b).